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A Manual for Eutrophication Control in the Subarctic

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Northern Affairs Program

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ABSTRACT

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A set of models suitable for the management of cultural eutrophication in the subarctic has been developed using data from a study (1977-1983) of 10 lakes located near Schefferville, Québec. The data suggest that, as in temperate lakes, the summer algal biomass and Secchi disc transparency are strongly correlated with summer concentrations of total phosphorus in the water column. These models, coupled with uncertainty analysis, are used to develop a stepwise procedure for the prediction of lake trophic state from phosphorus loading. An example based on published data for subarctic Harding Lake, Alaska is used to demonstrate and to verify the modeling procedure.

A Manual for Eutrophication Control in the Subarctic

As population pressures grow in Canada, so too will the risk of water quality degradation. In the Canadian north, resource exploration, development of new permanent settlements, and increasing recreational activity are creating new demands on northern water resources. It is important to manage this water in a way which will sustain fish and wildlife populations, and still accommodate other water uses such as boating, canoeing and domestic water supply (Anon. 1983).

One aspect of water quality which has received a great deal of attention is the nutrient enrichment, or eutrophication, of lakes and reservoirs in temperate latitudes. The most obvious manifestations of eutrophication are summer blooms of algae in the lake's surface waters, and the growth of algae and rooted plants in shallow nearshore areas (Ryding 1981). However, these algal blooms can in turn lead to reduced water transparency, taste and odour problems, and reduced oxygen levels in the lake water (cf. Reckhow and Chapra 1983; Persson et al. 1983). Empirical evidence (e.g. Sawyer 1947; Sakamoto 1966; Vollenweider 1968; Dillon and Rigler 1974a; Jones and Bachmann 1976) and experimental studies (Schindler et al. 1978) have confirmed that the principal cause of eutrophication is an increase in the supply of phosphorus to the lake from the watershed. These studies have in turn led to lake management strategies based on phosphorus control (cf. Dillon and Rigler 1975; Ryding 1981; Vollenweider and Kerekes 1980, 1982; Reckhow and Chapra 1983), and recent studies (e.g. Smith and Shapiro 1981a,b; Rast et al. 1983) have confirmed that phosphorus control is an effective management practice.

INTRODUCTION

However, despite the diversity of eutrophication models which have been developed and applied worldwide in warmer latitudes (e.g. Dillon and Rigler 1975; Cullen et al. 1978; Walmsley and Butty 1979; Vollenweider and Kerekes 1982; Thornton and Walmsley 1982), few general models yet exist which can be used in subarctic lake management (cf. Fox et al. 1979). The purpose of this paper is thus to provide a series of empirical models, which can be used for the rational control of cultural eutrophication in the subarctic.

Studies of nutrients and eutrophication in the subarctic

When this study was begun, only limited information was available on the response of subarctic lakes to nutrient enrichment. Symptoms of cultural eutrophication had been noted in Kam Lake, N.W.T. (Pick 1975; Bell et al. 1975), and in Big Trout Lake, Ontario (Jackson et al. 1978; Zimmerman et al. 1982a,b). However, an experimental study of the effects of nutrient additions in two subarctic Swedish lakes (Persson et al. 1977; Jansson 1978; Lundgren 1978; Persson 1978) clearly confirmed the dominant role of nutrients as regulators of algal productivity in subarctic waters.

The case of Big Trout Lake provides a good example of the effects of cultural eutrophication on subarctic communities. The lake is large (650 km^2) and relatively deep (mean depth, 10.7 m), and is located in the central Patricia region of Northern Ontario (Zimmerman et al. 1982b). It receives sewage from the nearby community of Big Trout Lake, which has led to the eutrophication of the bay surrounding the island on which most of the community's residents live. As noted by Zimmerman et al. (1982b), eutrophication of the bay was accompanied by all three symptoms mentioned previously: decreased transparency of the water, resulting in the loss of submerged navigational markers; nuisance algal growth, leading to taste problems with domestic water; and probable decreases in dissolved oxygen, which may be associated with movement of salmonid fishes out of the bay. These authors concluded that phosphorus control was the key to controlling cultural eutrophication in subarctic lakes such as Big Trout Lake, but also noted the lack of appropriate models on which management decisions could be based.

In response to this need, the late Frank Rigler began a project based in Schefferville, Québec (54°49'N, 66°50'W) in 1977 to study the response of subarctic Canadian lakes to nutrient enrichment. The results of the study (cf. Smith et al. 1984a,b) and recent developments in uncertainty analysis in lake water quality modeling (Reckhow and Chapra 1983) will be used below to develop detailed guidelines for the national management of cultural eutrophication in the subarctic.

An overview of the model concepts

A simplified scheme of lake response to phosphorus loading is given in Fig. 1. The phosphorus supply to lakes comes primarily from atmospheric sources (via snowfall, rainfall and dustfall) and from fluvial sources (via stream flow into the lake). The fluvial inputs may or may not include phosphorus from upstream point sources, such as sewage treatment plants or industrial waste inputs. If point sources exist, they may in some cases enter the lake directly. Fluxes of groundwater to the lake may provide additional phosphorus inputs, but this process is not yet well understood, and will not be considered here.

As can be seen in Fig. 1, the phosphorus supply to the lake and the rate of supply of water interact to determine the total phosphorus content of the lake water. A number of mathematical models exist which predict the mean phosphorus concentration (TP, $mg.m^{-3}$) from these two variables (cf. Dillon 1974; Vollenweider 1976; Reckhow and Chapra 1983).

In this study we have focussed on two principal models. The first (Eq. 1) was used by Dillon and Rigler (1975) in their method for predicting the capacity of southern Ontario lakes for development:

$$TP = \frac{L(1-R_p)}{q_s} \quad . \qquad (1)$$

In this theoretical model, L ($mgP.m^{-2}.yr^{-1}$) is the annual mass loading of total phosphorus to the lake from all sources, divided by the lake surface area. The phosphorus retention coefficient, R_p, is the

fraction of the phosphorus inputs which is retained by the lake each year.

The areal water loading to the lake (q_s, $m.yr^{-1}$) is calculated as the ratio of the annual water outflow from the lake (Q_{out}, $m^3.yr^{-1}$) to the lake surface area (A_o, m^2) (cf. Eq. 8). Thus, if L, R_p and q_s can be predicted, we can also predict the total phosphorus concentration in the lake, and we can quantify the changes in TP which we can expect when the phosphorus load is modified.

As noted by Dillon and Rigler (1974b), in order to use Eq. 1 properly we must also be able to make independent estimates of R_p from other easily measurable variables. Smith et al. (1984a) have tested 8 current empirical models predicting R_p from lake hydraulic characteristics and have concluded on the basis of prediction error that the model of Ostrofsky (1978a) is the most appropriate for subarctic lakes. However, as noted by Reckhow and Chapra (1979), error analyses of models predicting R_p must be made by re-analyzing the model development data set. We have made such an analysis (data not shown) and have concluded that the prediction error for Ostrofsky's (1978a) model is unacceptably large for subarctic lakes, which frequently have low areal water loading rates.

An alternative, empirically derived phosphorus loading model which has been used extensively in temperate regions is that of Reckhow and Simpson (1980):

$$TP = \frac{L}{11.6 + 1.2q_s} \qquad (3)$$

This model does not require the estimation of a phosphorus retention coefficient because, in effect, it is already estimated in the model (cf. Eq. 8.23 in Reckhow and Chapra 1983). An analysis of data from a study (Freund 1976) of three sewage-enriched lakes in Schefferville, Québec showed that the predictions of mean TP made by Eq. 1 and Eq. 3 were almost identical (data not shown). However, the large prediction error for phosphorus retention noted above made Eq. 1 unacceptable, and we have chosen Eq. 3 as the most suitable model for water quality management in the subarctic. The Vollenweider (1976) phosphorus loading model was not chosen because it assumes that phosphorus retention can be predicted well from the square root of the hydraulic residence time (cf. Larsen and Mercier 1976; Vollenweider 1976). An analysis of the Schefferville data (cf. Smith et al. 1984, Table 2) showed that phosphorus retention in these subarctic lakes was very poorly predicted from the Larsen and Mercier (1976) model.

In addition to the model predicting changes in the total phosphorus concentration in the lake, we also require models which link total phosphorus to the water quality variables which are of concern to users in the north. Although summer water temperatures are frequently too cool for swimming, there is also the potential for sport fishing, boating and canoeing, as well as domestic water use (Anon. 1983). In their method for predicting the capacity of southern Ontario lakes for development,

Dillon and Rigler (1975) used summer chlorophyll *a* concentrations (a measure of algal biomass) and summer Secchi disc transparency as their water quality criteria. We will retain these guidelines, but note that these criteria can be altered in the future if further northern research indicates that they are in error (Table 1).

Smith et al. (1984a) have tested the applicability of the Dillon and Rigler (1974a) and Smith (1982) models predicting summer chlorophyll *a*, and noted that their data from Schefferville lakes fit within the 95% confidence limits for the Dillon and Rigler (1974a) model:

$$\log \text{chl}a = 1.449 \log TP_{spring} - 1.136 . \qquad (4)$$

They also noted that the data could be predicted with even greater precision from the Smith (1982) model, which includes total nitrogen (TN) as an additional variable. However, the use of this model is not yet feasible in the subarctic because of a lack of suitable empirical nitrogen loading models. It was thus clear that total phosphorus was the most practical predictor of chlorophyll *a* in subarctic lakes, and a new model was developed based on the Schefferville data set (Fig. 2):

$$\log \text{chl}a = 0.975 \log TP - 0.559, \; r^2 = 0.94 , \qquad (5)$$
$$\text{root mean square error} = 0.1140.$$

In contrast to the Dillon and Rigler (1974a) model, which was based on measurements of total P at spring turnover, Eq. 5 was developed using summer mean (15 June - 15 September) TP values. It is also important to note that the initial model constraints (TN:TP ratio $>$ 12) used by Dillon and Rigler (1974) were relaxed in this analysis. We felt that it was most appropriate to model the response of subarctic lakes to sewage enrichment, which typically reduces the TN:TP ratio in lakes (cf. Ryding 1981).

In addition to the total phosphorus-chlorophyll *a* model, it was felt that an empirical model relating summer mean Secchi disc transparency (SD, m) to summer mean total phosphorus was also desirable. The Schefferville data set provided the following model:

$$\log SD = -0.578 \log TP + 1.182, \quad r^2 = 0.87 , \qquad (6)$$
$$\text{root mean square error} = 0.0949.$$

THE MODELING PROCEDURE

The procedure outlined below has been adapted from the procedures outlined by Dillon and Rigler (1975) and by Reckhow and Simpson (1980). In this section we will illustrate how lake data can be used to predict the water quality (trophic state) of subarctic North American lakes. Data from Harding Lake (Alaska) will be used to illustrate the use of this procedure.

Harding Lake (64°25'N, 146°50'W) is located near Fairbanks, Alaska. Although it is a large, deep lake, it has been heavily developed, with occupancy of private lots on nearly 75% of the lakeshore. Development is now proceeding on second-tier lots in many sections, and this development

has created concern for the water quality of the lake. This concern led to the studies by LaPerriere et al. (1978) which provide the data base for this analysis.

The method presented here will proceed in the following order: Step 1, estimation of areal water loading (q_s); Step 2, estimation of areal phosphorus loading (L); Step 3, calculation of predicted lake total P concentration; Step 4, estimation of phosphorus prediction uncertainty; Step 5, calculation of predicted summer chlorophyll *a* concentration; Step 6, calculation of chlorophyll *a* prediction uncertainty; and Step 7 (optional), calculation of predicted Secchi disc transparency.

Step 1: Estimation of q_s (Areal Water Loading)

The estimation of q_s involves the solution of Eq. 7 and 8:

$$Q_{out} = (A_d \cdot r) + A_o (P - E_v) \quad (7)$$

$$q_s = Q_{out}/A_o \cdot \quad (8)$$

where q_s = areal water loading ($m.yr^{-1}$)

Q_{out} = lake outflow water volume ($10^6 m^3.yr^{-1}$)

A_d = watershed area ($10^6 m^2$)

A_o = lake surface area ($10^6 m^2$)

r = total annual unit runoff ($m.yr^{-1}$)

P = total mean annual precipitation ($m.yr^{-1}$)

E_v = mean annual lake evaporation ($m.yr^{-1}$) .

Lake surface area should be determined from available information if possible; however, if these data are not available, the surface area ($10^6 m^2$) can be determined by planimetry of an aerial photograph of known scale. The drainage basin area (A_d) should be outlined on standard scale topographic maps or aerial photograph, and calculated by planimetry. Mean annual runoff (r, $m.yr^{-1}$) and mean annual precipitation (P, $m.yr^{-1}$) for locations in Canada and the United States south of 60°N can be estimated from tables given in Hare (1980). However, it should be noted that Hare (1980) suggests that runoff estimates in the Labrador-Ungava region may be in error by as much as 15%. For this region, readers may consult Findlay (1966) and Wright (1981) for relevant data. For other locations, these parameters may be interpolated from other references (e.g. Hare and Hay 1971; Hydrological Atlas of Canada 1978; Korzoun et al. 1977). Annual lake evaporation can be estimated in Canada from Morton (1983), and in the United States from U.S. Geological Survey measurements. Relevant hydrological references can also be found in Dingman (1973; 1975). However, it should be emphasized that, where possible, direct measurements of lake outflow are preferable to the estimates made using Eq. 7 and 8, since direct estimates of Q_{out} will decrease modeling uncertainty.

For Harding Lake, the effective drainage basin area has been estimated to be 2000 ha (LaPerriere 1981). The necessary calculations are:

a) $$Q_{out} = (20.0 \times 10^6)\ (0.188) + (9.88 \times 10^6)\ (0.305 - 0.180)$$
$$= 3.76 \times 10^6 m^3.yr^{-1}$$

b) $$q_s = (3.76 \times 10^6)/(9.88 \times 10^6)$$
$$= 0.381\ m.yr^{-1}$$

where mean annual runoff was estimated from Hare and Hay (1971), and lake evaporation was taken from LaPerriere et al. (1978).

Step 2: Estimation of L (Areal Phosphorus Loading)

The watershed of the lake should be divided on a topographic map into subunits for all inflows, and the use and area of each should be determined ($10^6 m^2$). The estimated total loading of phosphorus to the lake will be the sum of export from each subwatershed, inputs from point sources (such as sewage effluent), inputs from non-point sources (such as cottage septic tanks) and atmospheric deposition onto the lake surface. The total annual loading (J_T, $kg.yr^{-1}$) is thus:

$$J_T = J_f + (E_u \cdot A_{du}) + (E_{pr} \cdot A_o) + [\text{No. of capita-years} \times (1\text{-S.R.})] + PSI \qquad (9)$$

where:

J_f = annual phosphorus mass export from forested subwatersheds ($kg.yr^{-1}$), calculated from forested subwatershed area (A_{df})

E_u = annual phosphorus export coefficient for urban subwatersheds ($kg.10^{-6}m^{-2}.yr^{-1}$)

A_{du} = area of urban subwatersheds ($10^6 m^2$)

E_{pr} = annual bulk atmospheric deposition of total phosphorus to the lake surface ($kg.10^{-6}m^{-2}.yr^{-1}$)

A_o = lake surface area ($10^6 m^2$)

E_{st} = annual phosphorus export coefficient to septic tanks which may affect the lake ($kg.(capita - yr)^{-1}.yr^{-1}$)

No. of capita-years = No. of capita-years of occupancy in watershed in facilities serviced by septic tank/tile field systems which may affect the lake

S.R. = soil phosphorus retention coefficient (unitless)

PSI = point source phosphorus inputs ($kg.yr^{-1}$) .

Based on the procedure suggested by Reckhow and Simpson (1980), high, most likely, and low export estimates will be selected for each of the phosphorus sources given in Eq. 9. The high and low loading estimates represent additional loading error which must be added to the model error for the calculation of total prediction uncertainty. The choice of suitable high and low loading values may be difficult, since very little is known about phosphorus export characteristics in northern locations. However, a broad range between these two estimates is probably desirable since the characteristics of the lakes to which this procedure is applied may differ significantly from those which were used to develop the models presented here. We will present relevant data here but the reader should consult Uttormark et al. (1974) and Reckhow et al. (1980) for other relevant information on phosphorus export coefficients and their use.

The method for estimation of total annual phosphorus loading to the lake, using Eq. 9, will be presented below in a series of six substeps.

Step 2A: Estimation Of Phosphorus Export From Forested Watersheds

Assuming that most subarctic watersheds will be forested (no data for phosphorus export from subarctic tundras are yet available), the areas of each subwatershed can be used to estimate phosphorus exports from the land to the lake. Phosphorus export from 94 forested catchments worldwide has recently been analyzed by Prairie and Kalff (1984), who find the following relationship:

$$\log J_f = 0.986 \log A_{df} + 0.914, \quad r^2 = 0.93 \qquad (10)$$

where J_f is the annual phosphorus mass export from forested watersheds ($kg.yr^{-1}$) and A_{df} is watershed area ($10^6 m^2$). The model development data set included several subarctic watersheds studies by Brunskill et al. (1980), and thus it is felt that Eq. 10 can be used to estimate phosphorus export in northern regions given the absence of a broad subarctic data base.

In this substep, we will assume that the most likely phosphorus export estimate will be that predicted by the model, and that the upper and lower 95% intervals for the predictions can be used to estimate the high and low export values, respectively. This prediction interval may be calculated using the following equation (Steel and Torrie 1960):

$$95\% \text{ P.I.} = \hat{y} \pm 0.912 \sqrt{1.011 + [(x - 0.785)^2/252.4]} \qquad (11)$$

where $\hat{y}$ is the value (in $\log_{10}$ units) predicted by Eq. 10 from the value of A_{df} for the watershed in question and x is simply $\log_{10}$ (A_{df}) for the same watershed. The high and low (E_{fH} and E_{fL}) coefficient values are then estimated by taking the antilogarithm (base 10) of the 95% P.I. values. To correct for bias (Sprugel 1983), the most likely value for phosphorus export is calculated as:

$$J_{fML} = (\text{antilog } \hat{y}) \times 1.74 \; . \qquad (12)$$

For Harding Lake, the appropriate estimates of loading from this source (assuming that all of the watershed is forested) are thus:

$$A_{df} = 2000 \text{ ha} = 20 \times 10^6 m^2$$

$$\hat{y} = 2.1969$$

$$J_{fH} = 1301.3 \text{ kg.yr}^{-1}$$

$$J_{fML} = 273.8 \text{ kg.yr}^{-1}$$

$$J_{fL} = 19.0 \text{ kg.yr}^{-1}$$

Although the percentage of urban areas in subarctic watersheds is typically minimal, in certain cases (e.g. Yellowknife, N.W.T., and Schefferville, Québec) a calculation of urban phosphorus inputs is necessary. Because of a lack of relevant information, we simply suggest the use of the coefficents given in Table 2. It should also be noted here that if the purpose of the loading analysis is to predict the impact of development and changing land use, high and low estimates of future values for A_{df} and A_{du} are needed to reflect the uncertainty in land use projections (Reckhow and Simpson 1980). These authors also suggest that land area uncertainty and phosphorus export uncertainty should be paired in the following manner: the high estimate for area should be multiplied by the high value for phosphorus export. The most likely and low cases should be similarly paired, and the results should be included in the calculation of total loading (J_T) in step 2E. For Harding Lake, there are no urban inputs.

The contribution of atmospheric deposition to phosphorus loading can be quite high in certain regions. For example, in the Lakeshore Capacity Study (Ontario Ministry of the Environment 1983), the atmosphere was the single most important source of phosphorus for 4 of the 6 lakes studied in all four years (1976-1980). These results are consistent with other mass budgets from lakes on the Canadian Precambrian Shield (cf. Dillon and Reid 1981), and emphasize that atmospheric contributions to a lake will be most significant when the ratio of the watershed (A_d) to the lake area (A_o) is small.

A summary of bulk deposition values to Canadian watersheds is given in Table 3. However, Prepas (1979) notes that the two highest values (Dillon 1974, and Nicholls and Cox 1978) were obtained from land-based collectors. Gomolka (1975) found that measurements of bulk phosphorus deposition made from land-based collectors overestimated the values obtained from collectors based on a lake, and thus these values may not be representative. Note that the variability is still large, however, even between subarctic locations. The value obtained by Ostrofsky (1978a) at Yellowknife, N.W.T. (40 $\text{kg.10}^6\text{m}^{-2}\text{.yr}^{-1}$) is similar to the values obtained by the Ontario Ministry of the Environment (1983) near Dorset,Ontario. In contrast, the bulk deposition at Schefferville (8.2 $\text{kg.10}^6\text{m}^{-2}\text{.yr}^{-1}$) is much closer to the values obtained by Foster (1974) and Swanson (1976) in Southern Ontario. The basis for this variability is not yet known.

Based on high, most likely, and low values for E_{pr} (Table 2), the atmospheric inputs of phosphorus to Harding Lake are estimated as:

$$J_{prH} = (9.88 \times 10^6)\ (58.4) = 577\ kg.yr^{-1}$$

$$J_{prML} = (9.88 \times 10^6)\ (27.6) = 273\ kg.yr^{-1}$$

$$J_{prL} = (9.88 \times 10^6)\ (5.6) = 55.3\ kg.yr^{-1}$$

Step 2B: Estimation Of Capita-Years

Following Rodiek (1979), the number of septic systems which lie within 100 m of the lake shoreline, or within 100 m of an inflowing tributary, should be determined. The number of persons contributing to these septic systems must then be estimated and expressed as capita-years. When the goal of the lake analysis is to assess future conditions, high, most likely and low population estimates should be paired with high, most likely, and low per capita septic tank inputs (Table 2) to estimate septic tank phosphorus loading uncertainty (cf. Reckhow and Simpson 1980). The general equation used to calculate capita-years is:

$$\text{No. of permanently inhabited living units} \times \text{average No. of persons per living unit} \times \frac{\text{No. of days inhabited per unit per year}}{365} \quad (13)$$

plus

$$\text{No. of seasonal living units} \times \text{average No. of persons per living unit} \times \frac{\text{No. of days inhabited per unit per year}}{365}$$

In the case of Harding Lake, the developed area includes 262 lakefront lots, of which 80% have lake frontage of less than 30 m (LaPerriere et al. 1978). One living unit is assumed per lot, with a total of 500 seasonal occupants who spend 46 days during the summer at the cabins. In addition, during fiscal year 1973 a total of 50,636 people visited the Harding Lake recreational area for bathing, camping and picnics. However, waste from the recreation centre is stored in pumped vaults and shipped to Fairbanks for treatment (J.D. LaPerriere, University of Alaska, pers. comm.), and thus does not affect the lake. The calculation is thus:

		permanent		seasonal		
Total No. of capita years	=	0	+	(262)(1.91)(46/365)	=	63.1 capita-years

Step 2C: Estimation Of S.R. (Soil Phosphorus Retention Coefficient)

The impact of septic systems to lakes is potentially great because their effluent typically contains ca. 15 $mg.L^{-1}$ total phosphorus, a value which is 1,000 - 10,000-fold greater than that in undisturbed lakes (Otis and Boyle 1976). However, soils are typically efficient in retaining

phosphorus from septic systems, and several recent studies suggest that removal of more than 95% of effluent phosphorus is attained in most drainage fields (Jones and Lee 1977; Sommers et al. 1977; Gilliom and Patmont 1983), with lower retention being observed in older systems which are located in persistently saturated soils (Gilliom and Patmont 1983). However, in soils on the Canadian Precambrian Shield, Dillon and Rigler (1975) suggest that the shallow. coarse-textured sandy or muck soils have poor retention characteristics. In contrast, in areas such as Schefferville, Québec, the groundwater is very iron-rich (Drake and Freund 1981), and phosphorus retention by the soils probably exceeds 90-95% (T.R. Moore, McGill University, pers. comm.).

Because permanent septic systems are more likely to be the most common means of wastewater disposal in small subarctic communities, the use of three retention coefficients is desirable in this analysis. The choice of these coefficients should be made on the basis of a number of factors, including soil type, soil drainage, and soil permeability (cf. Reckhow et al. 1980). However, the choice of the coefficients in the end is probably a very subjective one, and thus leads to considerable loading uncertainty in cases where septic systems are potentially an important source of phosphorus for the lake. In this analysis, we will assume (J. LaPerriere, University of Alaska, pers. comm.) that phosphorus removal by soils around Harding Lake are typically very efficient, and have chosen the soil retention coefficients listed in Table 2. Table 2 also gives estimates of per capita phosphorus inputs to sewage systems around Harding Lake. The most likely value (0.8 kg $capita^{-1}.yr^{-1}$) is that used both in Dillon and Rigler (1975) and in the Lakeshore Capacity Study (Ontario Ministry of the Environment 1983).

Step 2D: Estimation Of PSI (Point Source Inputs)

In many subarctic communities, effluents from industry, sewage treatment plants, or other point sources flow directly into lakes. In the case of Schefferville, Québec, untreated sewage from the town flowed into Pearce Lake from 1953 until 1975, when a secondary sewage treatment facility was put into operation. In 1975, an estimated 6.6×10^3 kg of total phosphorus was discharged annually into the Pearce Lake chain from the sewage treatment plant, from industrial water use, and from untreated effluent from a housing development. It is important that such inputs be accurately quantified from direct measurements of effluent flow and total phosphorus content, where possible. If this is not possible, estimates of per capita phosphorus export from various degrees of sewage treatments (Table 4) can be multiplied by the number of persons serviced by the sewage system in order to estimate point source inputs. As in previous sections, a high, most likely, and low estimate of per capita phosphorus export should be coupled with corresponding estimates of future population size if predictions of future phosphorus loadings are desired.

In the case of Harding Lake, point source inputs of phosphorus were considered to be zero.

Step 2E: Calculation Of J_T (Total Phosphorus Mass Loading)

The calculation of the total mass of total phosphorus entering the lake is calculated from Eq. 9, using the data generated in the previous

four steps. For Harding Lake, the high, most likely and low values for J_T are calculated as:

$$J_{TH} = 1301 + 0 + 577 + (1)(63.1)(1 - 0.75) + 0$$

$$= 1301 + 0 + 577 + 15.8 + 0$$

$$= 1{,}894 \text{ kg.yr}^{-1}$$

$$J_{TML} = 273.8 + 0 + 273 + (0.8)(63.1)(1 - 0.95) + 0$$

$$= 273.8 + 0 + 273 + 2.5 + 0$$

$$= 549 \text{ kg.yr}^{-1}$$

$$J_{TL} = 19.0 + 0 + 55.3 + (0.3)(63.1)(1 - 0.99) + 0$$

$$= 19.0 + 0 + 55.3 + 0.19 + 0$$

$$= 74.5 \text{ kg.yr}^{-1} .$$

Step 2F· Calculation Of L (Annual Areal Phosphorus Loading)

To be used in the phosphorus model (Eq. 3), the results from the previous step must be expressed as loading per unit lake surface area ($\text{mgP.m}^{-2}\text{.yr}^{-1}$). The calculation is thus:

$$L = \frac{J_T . 10^6}{A_o} . \qquad (14)$$

For Harding Lake, we obtain the following three estimates:

$$L_H = \frac{(1{,}894 \text{ kg.yr}^{-1}) . (10^6 \text{ mg.kg}^{-1})}{9.88 \times 10^6 \text{m}^2}$$

$$= 192 \text{ mg.m}^{-2}\text{.yr}^{-1}$$

$$L_{ML} = \frac{549 \text{ kg.yr}^{-1}) . (10^6 \text{ mg.kg}^{-1})}{9.88 \times 10^6 \text{m}^2}$$

$$= 55.6 \text{ mg.m}^{-2}\text{.yr}^{-1}$$

$$L_L = \frac{(74.5\ kg.yr^{-1})\ .\ (10^6\ mg.kg^{-1})}{9.88 \times 10^6 m^2}$$

$$= 7.54\ mg.m^{-2}.yr^{-1}$$

Step 3: Calculation Of TP (Lake Phosphorus Concentration)

Estimates of the mean annual concentration of total phosphorus in the lake ($mg.m^{-3}$) can now be calculated from the three loading estimates calculated in Step 2. These estimates are calculated using Eq. 3 in the following manner:

$$TP_H = \frac{L_H}{11.6 + 1.2q_s} \quad (15)$$

$$TP_{ML} = \frac{L_{ML}}{11.6 + 1.2q_s} \quad (16)$$

$$TP_L = \frac{L_L}{11.6 + 1.2q_s} \ . \quad (17)$$

For Harding Lake, these calculations are:

$$TP_H = \frac{192}{11.6 + (1.2 \times 0.381)} = 15.9\ mg.m^{-3}$$

$$TP_{ML} = \frac{55.6}{11.6 + (1.2 \times 0.381)} = 4.6\ mg.m^{-3}$$

$$TP_L \quad \frac{7.54}{11.6 + (1.2 \times 0.381)} = 0.6\ mg.m^{-3} \ .$$

Step 4: Estimation Of Total Phosphorus Prediction Uncertainty

Because we have chosen to use Eq. 3 for the calculation of total phosphorus in subarctic lakes, the error analysis for these predictions is identical to that given in Reckhow and Simpson (1980). Following those authors, the uncertainty in TP will be calculated in 8 substeps.

Step 4A: Calculation Of log TP_{ML}

Take the logarithm of the most likely total phosphorus concentration, TP_{ML}, with the units expressed as $g.m^{-3}$. For Harding Lake:

$$\log TP_{ML} = \log(0.0046) = -2.337 . \quad (18)$$

Step 4B: Calculation Of s_{M^+} ("Positive" Model Error)

The model error (S_{mlog}, in logarithmic units) was previously found by Reckhow (1979) to be 0.128. To calculate "positive" model error for TP, we must add S_{mlog} to log TP_{ML}, and take the antilog of this sum. The difference between this value and TP_{ML} represents "positive" model error:

$$s_{M^+} = [\text{antilog } (\log TP_{ML} + S_{mlog})] - TP_{ML} . \quad (19)$$

For Harding Lake:

$$s_{M^+} = [\text{antilog } (-2.337 + 0.128)] - 0.0046 \ g.m^{-3}$$

$$= 0.0016 \ g.m^{-3}$$

$$= 1.6 \ mg.m^{-3} .$$

Step 4C: Caculation Of S_{m^-} ("Negative" Model Error)

To calculate "negative" model error for TP, subtract S_{mlog} from log TP_{ML}, and take the antilog of this value. The difference between this value and TP_{ML} is the "negative" model error:

$$s_{M^-} = [\text{antilog } (\log TP_{ML} - S_{mlog})] - TP_{ML} . \quad (20)$$

For Harding Lake:

$$s_{M^-} = [\text{antilog } (-2.337 - 0.128)] - 0.0046 \ g.m^{-3}$$

$$= -0.0012 \ g.m^{-3}$$

$$= -1.2 \ mg.m^{-3} .$$

The total uncertainty in TP is determined both by the model error (calculated above), and the loading error, which we will calculate below from the high and low estimates of TP which we obtained in Step 3.

Step 4D: Estimation Of s_{L^+} ("Positive" Loading Error)

The calculation of "positive" loading error is made by subtracting TP_{ML} from TP_H, and dividing the difference by 2:

$$s_{L^+} = \frac{TP_H - TP_{ML}}{2} . \quad (21)$$

For Harding Lake:

$$s_L^+ = (15.9 - 4.6)/2 = 5.65 \text{ mg.m}^{-3}$$

Step 4E: Estimation Of s_L^- ("Negative" Loading Error)

The calculation of "negative" loading error is made by subtracting TP_{ML} from TP_L, and dividing this difference by 2:

$$s_L^- = \frac{TP_L - TP_{ML}}{2} . \qquad (22)$$

For Harding Lake:

$$s_L^- = (0.6 - 4.6)/2 = -2.0 \text{ mg.m}^{-3} .$$

Step 4F: Estimation Of s_T^+ (Total "Positive" Prediction Error)

Total "positive" prediction error is calculated as the square root of the sum of the squares of "positive" model error and "positive" loading error:

$$s_T^+ = \sqrt{(s_M^+)^2 + (s_L^+)^2} . \qquad (23)$$

For Harding Lake:

$$s_T^+ = \sqrt{(1.6)^2 + (5.65)^2} = 5.9 \text{ mg.m}^{-3} .$$

Step 4G: Estimation Of s_T^- (Total "Negative" Prediction Error)

Total "negative" prediction error is calculated as the square root of the sum of the squares of "negative" model error and "negative" loading error:

$$s_T^- = \sqrt{(s_M^-)^2 + (s_L^-)^2} . \qquad (24)$$

For Harding Lake:

$$s_T^- = \sqrt{(-1.2)^2 + (-2.0)^2} = 2.3 \text{ mg.m}^{-3} .$$

Step 4H: Calculation Of the Prediction Interval for TP

Following Reckhow and Simpson (1980), we will calculate the prediction interval for TP from the total "positive" and total "negative" prediction errors estimated above. We will use a modifiication of the Chebyshev inequality (Benjamin and Cornell 1970), which can be expressed as:

$$\text{Prob}\left[TP_{ML} - hs_T^-) \leq TP \leq (TP_{ML} + hs_T^+)\right] \geq 1 - \left(\frac{1}{2.25h^2}\right) . \qquad (25)$$

Simply stated, this equation gives the probability that the true total phosphorus concentration in the lake lies within upper and lower bounds,

which are calculated from the total "positive" and total "negative" prediction errors. Substitution of a value of 1 for h gives 55% confidence limits for the value of TP, while values of h = 2 and h = 3 give roughly 90% and 95% confidence limits, respectively.

The level of uncertainty which a lake manager will accept will undoubtedly vary depending on the importance of the lake(s) in question. However, we will assume that a 90% level (and thus a 10% chance of error) is acceptable in the case of Harding Lake. We thus can calculate:

$$\text{Prob}\ [(TP_{ML} - 2s_{T^-}) \leqslant TP \leqslant (TP_{ML} + 2s_{T^+})] \geqslant 0.90\ .$$

For Harding Lake:

$$\text{Prob}\ [(4.7 - 2(2.3)) \leqslant TP \leqslant (4.7 + 2(5.9)] \geqslant 0.90,$$

or:

$$\text{Prob}\ [0.1 \leqslant TP \leqslant 16.5] \geqslant 0.90\ .$$

Thus, for Harding Lake, about 90% of the time that the prediction interval is estimated, we can expect the actual mean TP concentration in the lake to fall between 0.1 and 16.3 $mg.m^{-3}$. The summer mean concentration of TP in the lake was 14 $mg.m^{-3}$ (LaPerriere 1978), and our predictions are consistent with the value of TP which was actually observed.

Step 5: Calculation Of Summer Mean Chlorophyll Concentration

As was stressed earlier, two of the principal symptoms of eutrophication are the development of summer blooms of algae, and the reduction of water transparency. The summer biomass of algae is typically estimated by the concentration of chlorophyll a (chl$\underline{a}$, $mg.m^{-3}$), which can be predicted from TP_{ML} using Eq. 5:

$$\log \text{chl}\underline{a} = 0.975 \log TP - 0.559\ .$$

Like Eq. 12 in step 2A, the most likely value for chlorophyll (chl_{ML}) should be corrected for bias. However, the correction factor (1.03) is trivial, and, the calculation is thus:

$$chl_{ML} = \text{antilog}\ (\log \text{chl}\underline{a})\ . \qquad (26)$$

For Harding Lake:

$$\log \text{chl}\underline{a} = 0.975\ (\log 4.6) - 0.559$$

$$= 0.0872$$

$$chl_{ML} = \text{antilog}$$

$$= 1.2\ mg.m^{-3}\ .$$

Step 6: Calculation of Chlorophyll a Prediction Uncertainty

As in step 4, the prediction interval for the most likely mean summer chlorophyll a concentration (chl_{ML}) can be estimated using a modification of the Chebyshev inequality. The estimate of model error used in this procedure is the root mean square error (s_{clog}) for Eq. 5.

Step 6A: Calculation of log chl_{ML}

Take the logarithm of the most likely chlorophyll a concentration, chl_{ML}, with the units expressed as mg m^{-3}. For Harding Lake:

$$\log chl_{ML} = \log (1.2) = 0.0872 \qquad (27)$$

Step 4B Calculation of s_c+ ("Positive" Model Error)

The chlorophyll model error (s_{clog}, in log units) was found in this analysis to be 0.1140). As in step 4, "positive" model error can be estimated by:

$$s_c + = [\text{antilog } (\log chl_{ML} + s_{clog})] - chl_{ML} . \qquad (28)$$

For Harding Lake:

$$s_c+ = [\text{antilog } (0.0872 + 0.1140)] -1.2$$

$$= 0.389$$

Step 6C: Calculation of s_c- ("Negative" Model Error)

"Negative" model error (s_c-) is calculated in a similar manner

$$s_c- = [\text{antilog } (\log chl_{ML} - s_{clog})] - chl_{ML}. \qquad (29)$$

For Harding Lake:

$$s_c- = [\text{antilog } (0.0872 - 0.1140)] -1.2$$

$$= -0.260$$

Step 6D: Calculation Of the Prediction Interval for Chlorophyll

The 90% prediction interval for chla can be estimated from estimates of "positive" and "negative" model error calculated in the steps above:

$$\text{Prob } [(chl_{ML} - 2\ s_c-) \leqslant\ chl_{ML} \leqslant\ (chl_{ML} + 2s_c+)] \geqslant 0.90 .$$

For Harding Lake:

$$\text{Prob } [0.68 \leqslant\ chl_{ML} \leqslant\ 1.98] \geqslant\ 0.90$$

Thus, about 90% of the time, we can expect the most likely value of mean chlorophyll a in the lake to lie between about 0.68 and 1.98 $mg.m^{-3}$.

Step 7: Calculation Of Summer Mean Secchi Disc Transparency (optional)

The Secchi disc transparency can be predicted, if desired, from Eq. 6 for lakes which are known not to contain significant amounts of brown organic color. This was the case in Schefferville lakes from which Eq. 6 was developed, but in many other lakes organic color significantly reduces their transparency (e.g. Schindler 1971). In such cases, one could predict Secchi disc transparency using the relationship developed by Canfield and Hodgson (1983):

$$\ln SD = 2.01 - 0.370 \ln chla - 0.278 \ln C, \qquad (30)$$

where C is the organic color concentration in platinum units ($mg.L^{-1}$ Pt).

For Harding Lake, we can calculate the most likely Secchi disc transparency from the value of chl_{ML} estimated in Step 5. Assuming no color, we obtain from Eq. 6:

$$\log SD = -0.578 \log (4.6) + 1.182$$

$$= 0.799 .$$

Since the correction for bias for this equation is negligible, we calculate simply:

$$SD = \text{antilog} (\log SD)$$

$$= \text{antilog} (0.794)$$

$$= 6.3 \text{ m}$$

Summary of results for Harding Lake

Based on the calculations which we have made in the previous steps, we conclude that the most likely summer chlorophyll a concentration in Harding Lake is 1.2 $mg.m^{-3}$, with a 90% prediction interval of 0.68 - 1.98 $mg.m^{-3}$, and that the Secchi disc transparency is about 6 m. From Table 1, we also conclude that winter oxygen depletion under the ice and the risk of winter fishkill are minimal.

Water quality measurements in Harding Lake (LaPerriere et al. 1978) suggest a summer mean total phosphorus concentration of 14 $mg.m^{-3}$ and chlorophyll a concentration of 0.73 $mg.m^{-3}$. Both values lie within the 90% prediction intervals for TP and chla, and the observed summer mean Secchi disc transparency (11 m) exceeds the predicted value. Winter oxygen concentrations under the ice just prior to ice break-up in late May exceeded 11 $mg.L^{-1}$ in the top 20 meters, confirming that winter oxygen depletion was indeed negligible. We thus conclude that the procedure outlined here reasonably describes the observed conditions in Harding Lake.

Conclusions

The procedure which we have outlined above is intended to be a guide for the management of natural subarctic lakes with regard to cultural eutrophication. The method is relatively simple, requiring only a knowledge of the population and land use in a watershed, watershed area, annual runoff, and lake area. The fact that the model appears to make reasonable predictions of water quality in Harding Lake, Alaska - a lake much different in morphometry, geology, and geographical location from the lakes in the model development data set - suggests that this procedure should be suitable for use with a wide variety of subarctic lakes in North America. However, further tests of the method are desirable and it may be that certain aspects (e.g. phosphorus export coefficients, or the phosphorus loading model) will be changed in the future to reflect new information.

We also stress that this procedure is relevant primarily for the management of natural lakes, rather than impoundments. Models for the prediction of the trophic state of northern reservoirs are being developed (e.g. Ostrofsky 1978b; Grimard and Jones 1982), but have not yet been adequately tested. Furthermore, the impoundment of reservoirs can create a new set of problems which is not related to eutrophication alone. For example, the Churchill-Nelson River diversion in northern Manitoba has led to elevated mercury levels in the fishes of lakes affected by the diversion, and to a three-fold reduction in commercial whitefish catches in Southern Indian Lake (Bodaly et al. in press). Bodaly et al. (in press) also concluded that many of the major environmental effects of flooding and river diversion were either poorly predicted or not predicted at all in this system, despite extensive pre-development studies. There is thus a clear need for models and rational guidelines for the development of subarctic impoundments.

Even among natural lakes, however, we emphasize that eutrophication is only one of many closely-linked problems relating to land development in the north (cf. Eedy 1974). In particular, wastewater from northern communities contains not only phosphorus and nitrogen, but also high concentrations of pathogens and potentially toxic substances. In the cases of Kam Lake, N.W.T. (Bell et al. 1975) and Big Trout Lake, Ontario (Zimmerman et al. 1982a,b), dangerously high levels of sewage-derived bacteria made the water unfit for human consumption without disinfection. We thus urge that the eutrophication management guidelines presented here be integrated into a broader management policy which recognizes other user-related problems.

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Table 1 Maximum permissible summer average chlorophyll *a* concentrations (modified from Dillon and Rigler 1975).

Trophic State	Chlorophyll a, $mg.m^{-3}$	Lake characteristics
Oligotrophic	≤ 2	Low productivity, with summer mean Secchi disc readings > 5 m (except in colored or turbid lakes) Winter oxygen depletion under the ice and the risk of winter fishkill will be minimal.
Mesotrophic	5	Less aesthetically pleasing, with the possibility of nuisance algal growth. Summer Secchi disc transparency will be reduced to 2-5 m, and winter oxygen depletion will be elevated. Some movement of cold-water fishes (e.g. lake trout) away from more eutrophic embayments may occur.
Eutrophic	≥ 10	Nuisance algal blooms common, with low summer Secchi disc transparency (< 2 m). Winter oxygen depletion will be high, leading to probable losses of desirable fish species and a potential winterkill hazard in shallow lakes.

Table 2 Estimated inputs of phosphorus and soil phosphorus retention coefficients for Harding Lake, Alaska (based on data from Reckhow et al. 1980; Reckhow and Simpson 1980 and Table 3).

Source	Units	High	Most likely	Low
Urban (E_u)	$kg.10^6m^{-2}.yr^{-1}$	150	90	50
Atmospheric deposition (E_{pr})	$kg.10^6m^{-2}.yr^{-1}$	58.4	29	5.6
Septic tank inputs (E_{st})	$kg.capita^{-1}.yr^{-1}$	1.0	0.8	0.3
Soil Retention Coefficient (S.R.)	-	0.75	0.95	0.99

Table 3 Measurements of annual bulk atmospheric total phosphorus deposition in temperate and subarctic Canadian watersheds. Note year-to-year variability.

Location	Year	Bulk deposition $kgP.10^6.m^{-2}.yr^{-1}$	Reference
Southern Ontario	1967	35.0	Schindler and Nighswander 1970
Southern Ontario	1974-1975	36.9	Gomolka 1975
Haliburton, Ontario	1971-1972	77.2**	Dillon 1974
Southern Ontario	1975-1976	44.3*	Cross 1977
Experimental Lakes Area, Ontario	1970	29	Schindler et al. 1976
Experimental Lakes Area, Ontario	1972	24	" "
Experimental Lakes Area, Ontario	1973	24	" "
Muskoka-Haliburton, Ontario	1976-1977	58.4	Ontario Ministry of the Environment 1983
Muskoka-Haliburton, Ontario	1977-1978	36.5	" "
Muskoka-Haliburton, Ontario	1978-1979	30.2	" "
Muskoka-Haliburton, Ontario	1979-1980	21.8	" "
		5.6	Swanson 1976
		9.0	Foster 1974
		20.1	Kramer 1976
		35.0	Scheider 1978
	1973-1974	74.4**	Nicholls and Cox 1978
Yellowknife N.W T.	1975	40.0	Ostrofsky 1978a
Schefferville Québec	1976-1977	8.2	This study
	average	27.6***	

* underestimate; based on March - December measurements
** probably overestimated; see text for details
*** excluding three questionable values

Table 4 Estimates of per capita phosphorus loadings from sewage of various levels of treatment (after Reckhow et al. 1980).

Treatment	Location	Mean or median loading $kg.capita^{-1}.yr^{-1}$	Range $kg.capita.yr^{-1}$	Reference
None (raw)	Yellowknife, N.W.T.	0.55	-	Pick 1975
None (raw)	Resolute Bay, N.W T.	1.07	-	Schindler et al. 1974
Activated sludge	U.S.A.	0.89	0.32-4.99	U.S. EPA 1974
Trickling filter	U.S.A.	1.10	0.39-5.44	U.S. EPA 1974
Phosphorus removal	U.S.A.	0.57	0.23-1.81	U.S. EPA 1974
Primary settling and digestion	U.S.A.	0.82	0.27-3.18	U.S. EPA 1974
Oxidation pond	U.S.A.	1.07	0.36-3.63	U.S. EPA 1974
Sand filter	U.S.A.	2.86	0.77-6.11	U.S. EPA 1974

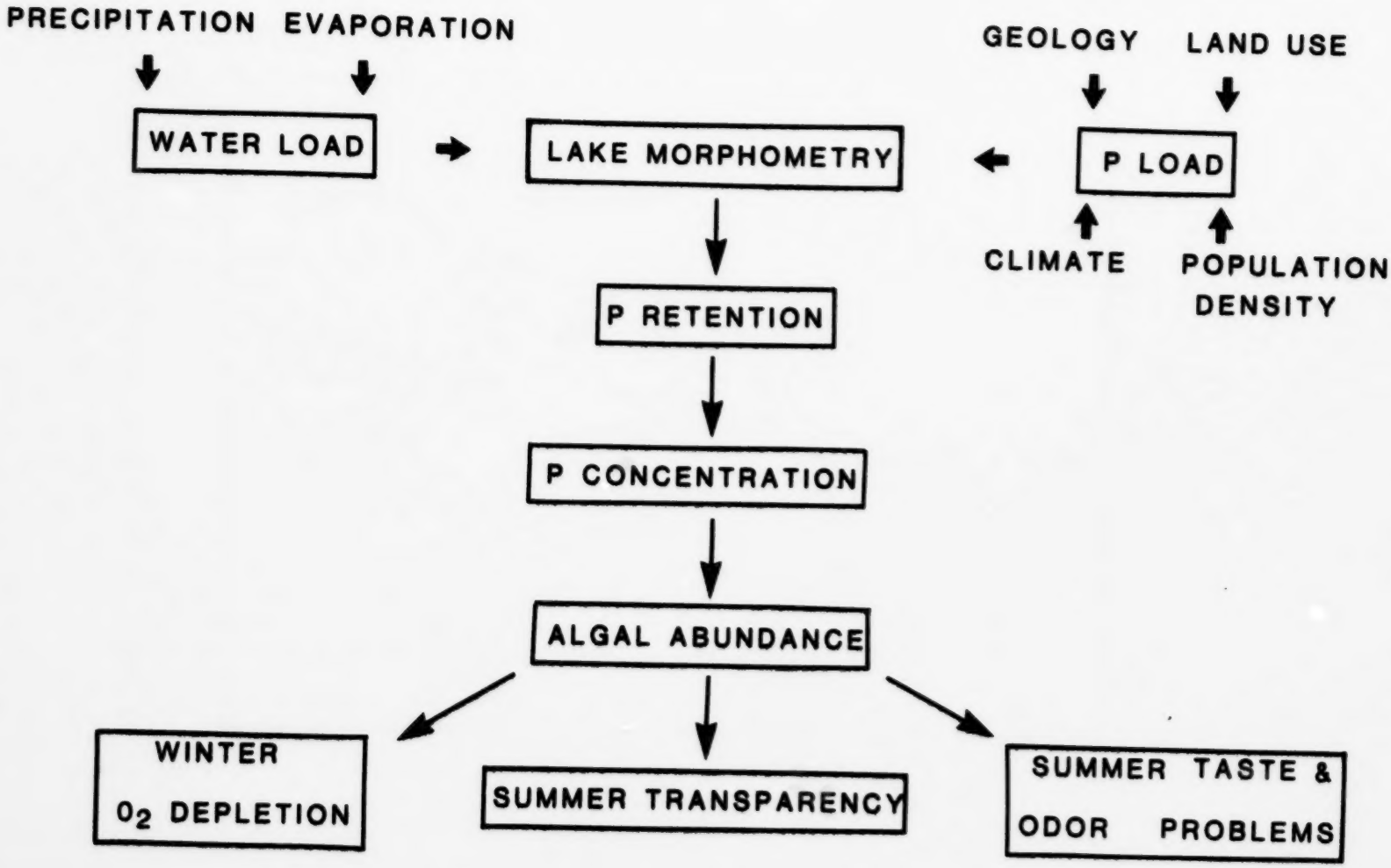

FIG. 1. SIMPLIFIED SCHEME OF LAKE RESPONSE TO PHOSPHORUS LOADING.

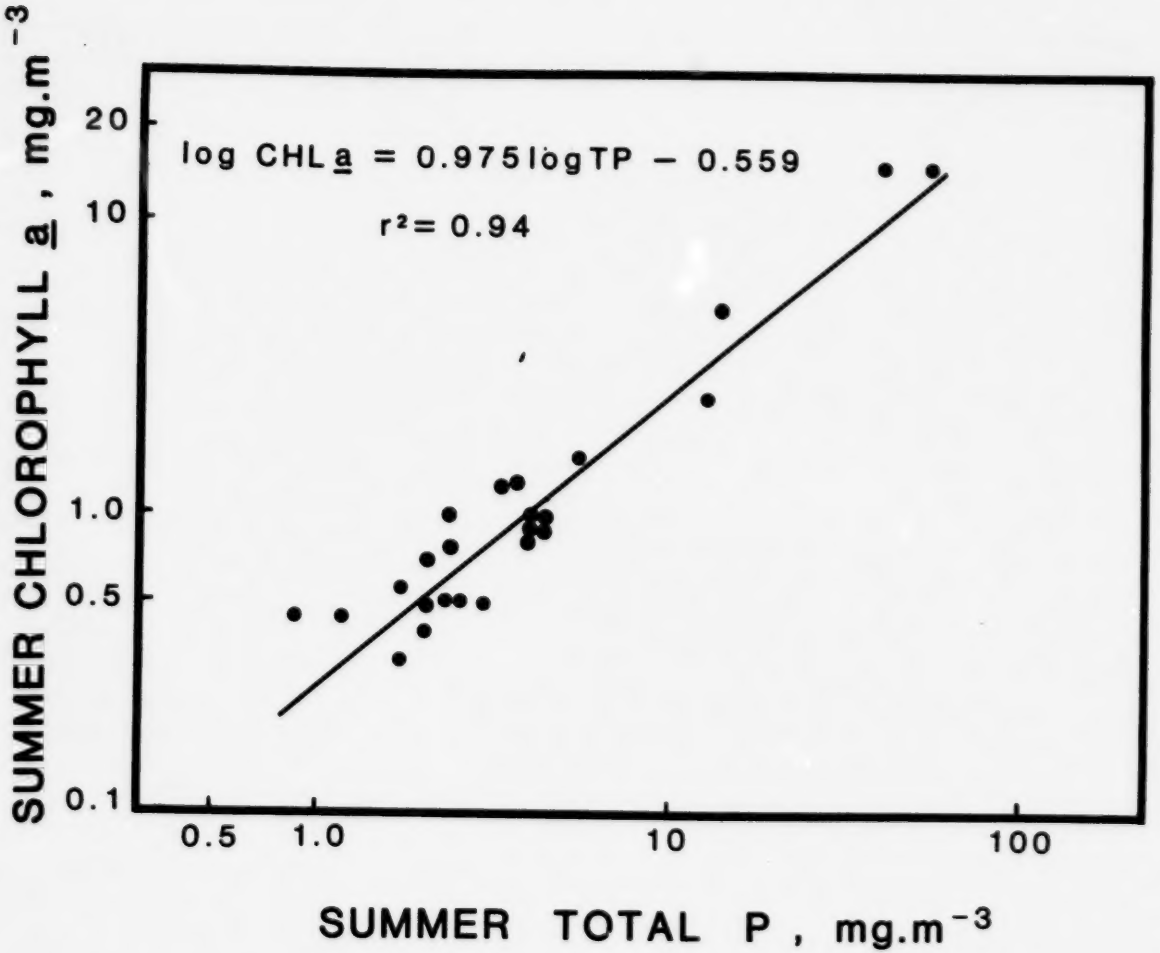

FIG. 2. RELATIONSHIP BETWEEN SUMMER MEAN CHLOROPHYLL a AND SUMMER MEAN TOTAL PHOSPHORUS IN 10 SCHEFFERVILLE LAKES.